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MODELING RISK OF HUNTING PRESSURE ON LARGE MAMMALS IN NORTHWESTERN ARGENTINA

By

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B.A. Environmental Studies, University of California Santa Barbara, Santa Barbara, California, 2003

Thesis

presented in partial fulfillment of the requirements for the degree of

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CONTENTS

THESIS FORMAT

I wrote this thesis formatted for submission to the journal *Biological Conservation*. Because of collaboration with another researcher, co-author will be listed as A. Moreno, I use the collective "we" throughout the thesis.

Baker, Loretta, M.S., December 2011 Resource Conservation

MODELING RISK OF HUNTING PRESSURE ON LARGE MAMMALS IN NORTHWESTERN ARGENTINA

Chair: Perry J. Brown

ABSTRACT

The subtropical Yungas and Chaco forests of northwestern Argentina are two of the most biodiverse and threatened biomes in South America. It is unclear how development pressure and increased human presence may be affecting wildlife in this increasingly fragmented and degraded landscape. We initiated a broad-scale analysis of the spatial distribution and magnitude of anthropogenic factors that may influence large mammal mortality due to potential human hunting pressure in a landscape linkage connecting these threatened forests. We conducted a literature review of Neotropical study sites that reported hunting of large mammals by indigenous people or colonists, and used this information to inform development of a risk distribution model. We identified linear distance values that represented the spatial patterns of hunter travel distance (i.e., willingness to travel) when in search of large bodied (>10 kg) prey species. To parameterize our model, we used information on percent forest cover, and values that reflect hunter travel distances as a function of distance from disturbance on the landscape, referencing roads and human settlements. The resultant risk map highlights gradients of risk of human-caused mortality due to hunting of large mammals potentially inhabiting or moving through the study region. We report patterns across the study landscape that show areas of relatively low mortality risk and putative linkages, while in other locations we report clear aggregations of high risk values suggesting areas of conservation concern. Where existing protected areas are close to or overlapping high risk areas, land managers should implement focused anti-poaching campaigns and prevent land clearing activities that could elevate human-caused risk of mortality. Likewise, locations at low risk of human-caused hunting mortality (especially those areas located amid the protected area network) may be robust for conservation, and thus should be considered a management priority. Minimizing new human disturbance, particularly in locations we report as low to moderate risk, should be actively pursued before these locations become targets of future land-use change. If managers seek to sustain the region's wildlife populations for future generations, then focused hunting control action and public awareness campaigns combined with forest conservation programs should be a high priority on the management agenda. Special funds are needed to improve managers' ability to control poaching throughout this region and help support new wildlife population studies to further focus conservation planning.

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1. INTRODUCTION

Human induced habitat loss and associated forest fragmentation are the leading cause of mammalian extinctions across the tropics (Wilkie et al. 2011), while unsustainable hunting represents the second most serious threat to mammals (Redford 1992, Bodmer et al. 1997, Cullen et al. 2000, Peres and Lake 2001, Mockrin et al. 2011). Fragmentation of species' habitat and hunting are tightly linked. Where extractive industries (e.g., logging, oil exploration) bring human settlements and the expansion of road networks into native and continuous tracts of forest, human access to once remote locations is enabled, thus increasing the magnitude of hunting pressure (Seijas 2004, Wilkie et al. 2011). Increased human disturbance to an area can disrupt species dispersal forcing individuals to navigate novel environments with landscape features that may threaten species distribution or persistence (Gardner and Gustafson 2004). As forest fragments become progressively more vulnerable to hunting pressure, recolonization rates from nonharvested source populations diminish, leading to reduced genetic exchange among species' populations (Bodmer et al. 1997, Peres and Lake 2003). Unsustainable wildlife harvest can lead to smaller effective population sizes, species range contractions, and the onset of inbreeding depression—all paths that can lead to species' extirpation or extinction (Mills and Allendorf 1996, Cushman et al. 2006, Rabinowitz and Zeller 2010).

Given that hunting is a generally diffuse and invisible activity, ascertaining the level of hunting pressure in a region, and its sustainability, is difficult (Bodmer et al. 1997). As reflected by the 'empty forest syndrome' the mere presence of forested landscapes provides little indication of the condition or vigor of wildlife populations within (Redford 1992, Bonaudo et al 2005, Wilkie et al. 2011). Understanding the spatial distribution of hunting pressure and its effects on wildlife population dynamics is necessary to assess hunting sustainability (Mockrin et al. 2011). Whether a given area has sufficient wildlife population numbers to allow sustainable offtake requires knowledge of the distribution, abundance, and growth rates of target species (Yackulic et al. 2011). Yet equally as important is the need for information on the distribution and magnitude of anthropogenic threats that influence risk of mortality for wildlife (Yackulic et al. 2011).

Studies from multiple tropical forest sites in Latin America show that hunting pressure, defined here as risk of human-caused mortality for mammals, can be spatially approximated from points of human access such as villages, ranch settlements, roads, and forest clearings (Redford 1992, Di Bitetti et al. 2008, Rabinowitz and Zeller 2010, Schwartz et al. 2010, Yackulic et al. 2011). Because humans are central place foragers, the probability of occurrence and the relative densities of exploited species increase with distance from human access points; although actual values at which species densities reach undisturbed levels vary greatly across species (Chiarello 1999, Novaro et al. 2000, Mockrin et al. 2011, Yackulic et al. 2011, Wilkie et al. 2011). Numerous researchers have reported the distance hunters travel from areas of human disturbance in search of wild game, which is one measure of mortality risk (Peres 2001, Altrichter and Boaglio 2004, Bonaudo et al. 2005, Altrichter 2006, Rabinowitz and Zeller 2010, Thoisy et al. 2010). Mortality risk is also approximated according to the relative risk that is attributed to different types of access (i.e., relative frequency and intensity of human use).

Areas surrounding villages are associated with very high mortality risk for mammals due to high human populations, and because hunting trips typically originate directly from villages into surrounding forested areas (Altrichter and Boaglio 2004, Altrichter 2005, Altrichter 2006, Rabinowitz and Zeller 2010). Ranch settlements are also high-risk locations for mammals, given the wide range of human activities that occur around most rural ranch sites (e.g., tending to livestock, obtaining water and firewood). In these circumstances, ranchers often opportunistically hunt as they encounter game (Altrichter and Boaglio 2004). However, relative risk associated with ranches is less than that of villages because human density is much lower on ranch sites. Roads are key access points for hunters seeking wild game. Relative frequency of use and road conditions can help determine the mortality risk mammals face. Because primary roads (including secondary roads) experience frequent human use and are generally easier to navigate (e.g., mobilize hunter activities), proximity to primary roads represents high mortality risk for mammals (Seijas 2004, Thoisy et al. 2010). Tertiary roads are generally less frequented by humans and located in more remote locations where accessibility is difficult (Seijas 2004, Franzen et al. 2006, Thoisy et al. 2010). Thus tertiary roads present lower relative mortality risk than primary roads. While hunters often report higher biomass return in high forest cover settings, lower forest cover settings offer more favorable conditions for hunters to view game because hiding cover for prey is limited and human access is improved (Parry et al. 2009, Mockrin et al. 2011). Low forest cover environments introduce risk of mortality for mammals, but this risk is not inherently high, unless low forest cover conditions are accompanied by risk factors such as proximity to roads and human settlements (Peres 2001, Naughton-Treves et al. 2003, Altrichter and Boaglio 2004, Parry et al. 2009).

Due to their large body size and widespread hunter preference for their meat, large mammals (10-160 kg) are particularly targeted by hunters (Redford 1992, Di Bitetti 2008, Paviolo et al. 2009, Sampaio et al. 2010). Large mammals are usually the first to disappear from an area experiencing light to moderate harvest (Cullen et al 2000, Di Bitetti 2008, Paviolo 2009). This is especially true for harvest-sensitive species including the lowland tapir (*Tapirus terrestris*) and white-lipped peccary (*Tayassu pecari*), and heavily persecuted species such as the jaguar (*Panthera onca*; Altrichter 2005, Altrichter et al. 2006, Rabinowitz and Zeller 2010). Species fecundity rates play a role in harvest sensitivity in cases where harvest rates exceed the ability of the species to reproduce, such as is the case for lowland tapir and jaguar (Bodmer et al. 1997). Some species have low tolerance for harvest due to other life history traits (e.g., white-lipped peccary herds group closely together when hunted, thereby allowing multiple individuals to be killed at one time; Sowls 1997). Hunting impacts on large mammals are problematic in areas of "pristine" forest where there is light harvest (Peres and Lake 2003), and consequences may be more serious in areas with increasingly isolated forest fragments, where hunter accessibility is expanding (Chiarello 1999).

My study focuses on a rural 20,000 km^2 forested connection between the Yungas and Chaco forests of the Salta and Jujuy region of Argentina, where exploitative development including oil exploration and agriculture has supported land use change that may present mortality risk for large mammals due to human hunting pressure (Grau et al. 2008, Gasparri and Grau 2009). In this region, risk of large mammal mortality may translate to bushmeat harvest and the potential that harvest levels are unsustainable (Bodmer and Robinson 2004, Altrichter 2005, Altrichter et al. 2006, Rabinowitz and Zeller 2010). Recent research suggests that increased hunting pressure is the most important and proximate cause of the decline of wildlife species population numbers and the contraction of species range in the region (Altrichter and Boaglio 2004, Altrichter 2006, Altrichter et al. 2006, Ojeda et al. 2008, Chalukian et al. 2009). This is considered to be particularly the case for large mammals known to inhabit this area including the giant armadillo (*Priodontes maximus*), white-lipped peccary, Chacoan peccary (*Catagonus wagneri*), collared peccary (*Tayassu tajacu*), giant anteater (*Myrmecophaga tridactyla*) capybara (*Hydrochoerus hydrochaeris*), red brocket deer (*Mazama americana*), lowland tapir, and jaguar (Peres and Lake 2003, Altrichter 2006, Ojeda et al. 2008). Presently, almost all hunting of native wildlife is prohibited according to provincial law in Salta and Jujuy, with few exceptions that allow the seasonal take of small mammals and select bird species (Resolution 142-10, Seasonal Sport Hunting Regulations, Salta Ministry of the Environment and Sustainable Development). Despite these prohibitions, the hunting of large native mammals is widely practiced and socially acceptable in the Yungas and Chaco forests (N. Politi, Fundación CEBio, personal communication).

Both the Yungas and Chaco forests are heavily influenced by historic and new development including agriculture, logging, oil exploration, and road construction, which has caused significant modification of the structure and function of these forest ecosystems (Tabeni et al. 2004, Talamo and Caziani 2003, Bonaudo et al. 2005, Grau et al 2005, Boix and Zinck 2008). Though a forested connection still exists in a landscape linkage located amid these subtropical forests, a dense road network, widespread human settlements, and newly planned deforestation zones suggest a problematic outlook for the persistence of wildlife in this region.

According to studies throughout Latin America, most hunting is unmanaged and harvest levels are often above those that can be sustained (Bodmer et al. 1997, Peres 2001, Bodmer and Robinson 2004). In the 20,000 km^2 study landscape conservation biologists are concerned about the future of large mammal persistence given the paucity of data on wildlife populations and uncertainty regarding large mammal hunting pressure. Given that detailed studies in the field are severely constrained by limited resources and time, my objective was to examine the distribution and magnitude of anthropogenic threats that may influence large mammal mortality risk. This study is the first attempt at an explicit spatial analysis of large mammal mortality risk due to human-caused factors across this landscape.

2. METHODS

2.1 Study area

2.1.1 Biophysical setting

The subtropical Yungas and Chaco forests constitute two of the most biodiverse and threatened biomes in South America (Ojeda et al. 2003). These forests represent an important interface between tropical and temperate biota, and harbor over 50% of Argentina's endangered species (Ojeda et al. 2003). Rich floral and faunal diversity provide ecological services including flood and erosion control, and provide local economic stimuli through logging and non-timber forest product production, supplying the region's food and fiber.

The 20,000 km² study landscape, delineated by the Argentine non-profit Fundación CEBio, was established based on the largely contiguous forested connection between the Yungas and Chaco forests, in the Salta and Jujuy Provinces of northwestern Argentina (Fig. 1). This landscape linkage represents the one of the region's last significant forested conduits, allowing the flora and fauna to move amid the forests (Figs. 1, 2; Rabinowitz and Zeller 2010). Forest connectivity is threatened by agricultural development pressure, particularly within the central portion of the linkage, creating a potential ecological bottleneck between the forests. Here there is a noticeable narrowing of the northern and southern sections of the linkage, reflecting the presence of extensive, nearly uninterrupted agricultural plantations that average 20-50 km in width (hence these locations were excluded in study area delineation). The central to eastern portions of the landscape linkage are predominately Chaco forest. This area is generally flat, with elevation gradients ranging between 200-500 m. Moving westward, elevation increases to 500-650 m in the transition zone between the Yungas and Chaco forests, reaching upwards of 1,000-2,500 m in the Yungas forest located in the westernmost portion of the study area (Brown et al. 2001, Boix and Zinck 2008). Natural vegetation changes gradually from east to west due primarily to changing climatic and topographic conditions (Brown et al. 2001, Altrichter 2006, Boix and Zinck 2008). This region supports agriculture, ranging from sugar cane and citrus in the west, to grain products such as soybean, wheat, and corn in the central and eastern portions. Cattle grazing, logging, and oil infrastructure development are evident throughout the landscape.

The Chaco is a vast plain extending across parts of Argentina, Bolivia, and Paraguay (46% of which is found in Argentina; Redford et al. 1990, Altrichter 2006). Approximately 70% of the study area is within the Chaco biome located east of the foothills of the sub-Andean mountain ranges in the climatic fringe between sub-humid and semiarid. Annual rainfall is between 650-880 mm, with seasonal flooding occurring in lower lying areas. Dryer years can be marked by as little rain as 150 mm, and wetter years approach 1,500 mm (Boix and Zinck 2008). The climate is markedly seasonal with most rainfall occurring between October-April . Average annual temperatures range between 18-20˚C, with temperatures exceeding 40˚C during the months of December through February. The Chaco forest consists of semi-deciduous thorny forests, dry thorny forests, open forests, palm savannas, and grasslands (Talamo and Caziani 2003). Vegetation is medium to tall xerophilous forest with many types of cacti and terrestrial bromeliads (Altrichter and Boaglio 2004). Dominant vegetation includes quebracho blanco (*Aspidosperma quebracho blanco)*, quebracho Colorado (*Schinopsis quebracho-colorado*), floss silk tree (*Chorisia speciosa)*, Guayacán negro (*Caesalpinea paraguariensis*), and mesquite (*Prosopis spp*.). The study area consists of mostly secondary forest (N. Politi, Fundación CEBIO, personal communication). Historically the Chaco was parkland or savanna consisting of patches of hardwood interspersed with grasslands. However, the combination of deforestation and intense overgrazing, timber harvest, and charcoal production is turning large areas of the

Chaco (including portions of the study area) into dense shrubland along with progressive erosion and desertification (Altrichter and Boaglio 2004, Gasparri and Grau 2009).

The western portion of the study area, mostly a transition zone from the Chaco to Yungas biome, consists of subtropical forest that can only be found on the eastern slopes of the Andes Mountains. The Yungas, with its northern extent originating in Venezuela, is a narrow band of humid forest that forms an ecologically diverse transition zone between the high Andean peaks west of the study area, and the semi-arid Chaco forest located along its eastern extent (Brown et al. 2001, Ojeda et al. 2008). Precipitation may reach up to 2,300 mm, with the majority of rainfall occurring between elevations of 1,000-1,500 m. Rainfall is concentrated (80-90%) between December-March (Grau and Brown 2000). Temperatures in this region can drop for short periods of time below 0˚C, but normally fall within 18-20˚C (Grau and Brown 2000). Dominant vegetation found in this region includes palo armarillo (*Phyllostylon rhamnoides*), palo blanco (*Calycophyllum multiflorum*), urundel (*Astronium urundeuva*) and cebil (*Anadenanthera colubrine*) (Grau et al. 2005). The Yungas forest in this area is confronted with significant extraction pressure for grazing, cultivation, logging, and other land clearing activities.

The study area is bordered along most of its northern boundary by the Bermejo River and associated tributaries. Two provincial protected areas are located within the landscape linkage

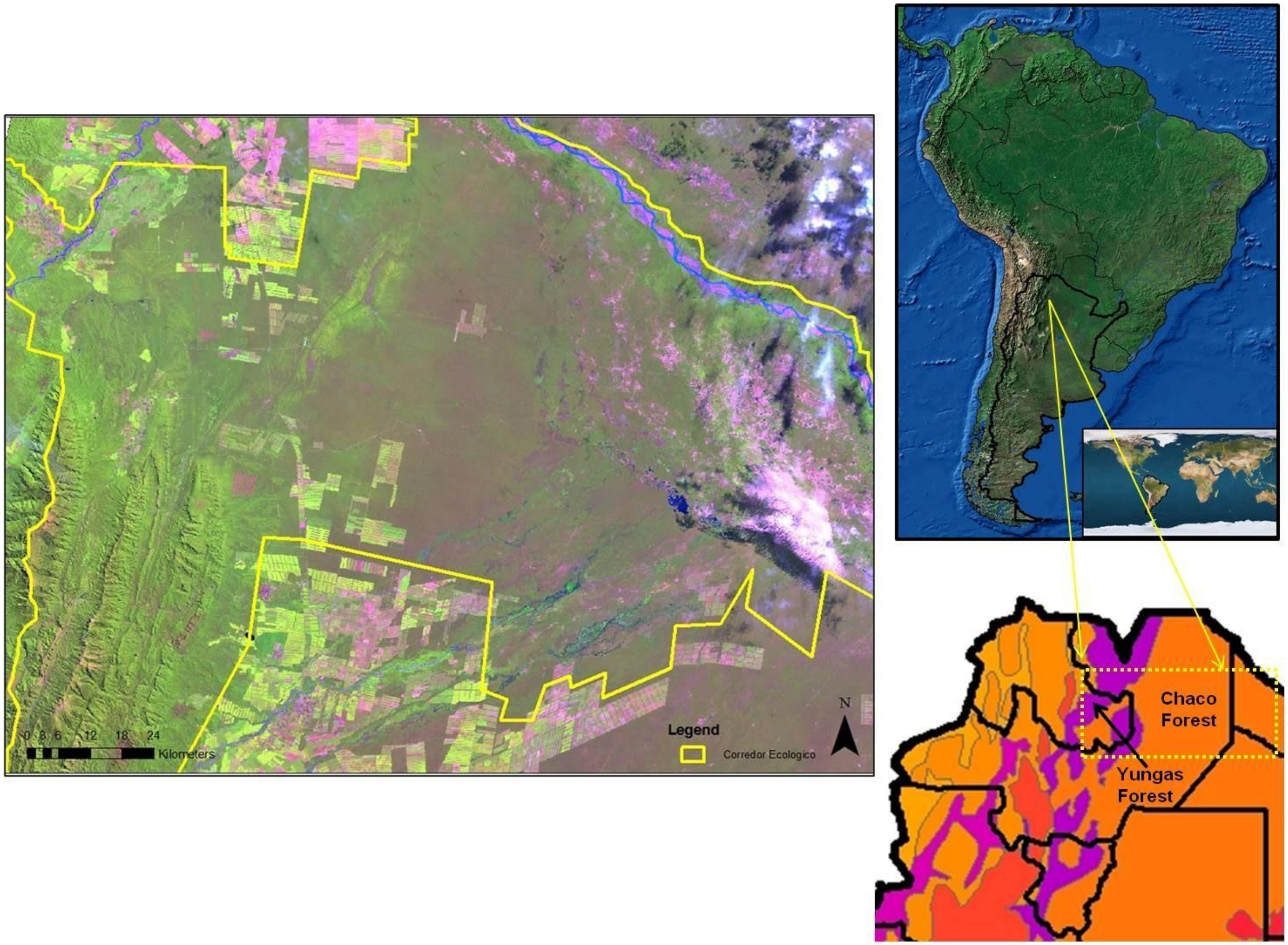

Figure 2: The study landscape linkage, outlined in yellow, located in the Salta and Jujuy Provinces of northwestern Argentina. This linkage, approximately 20,000 km², connects the subtropical Yungas and Chaco forests. 2011.

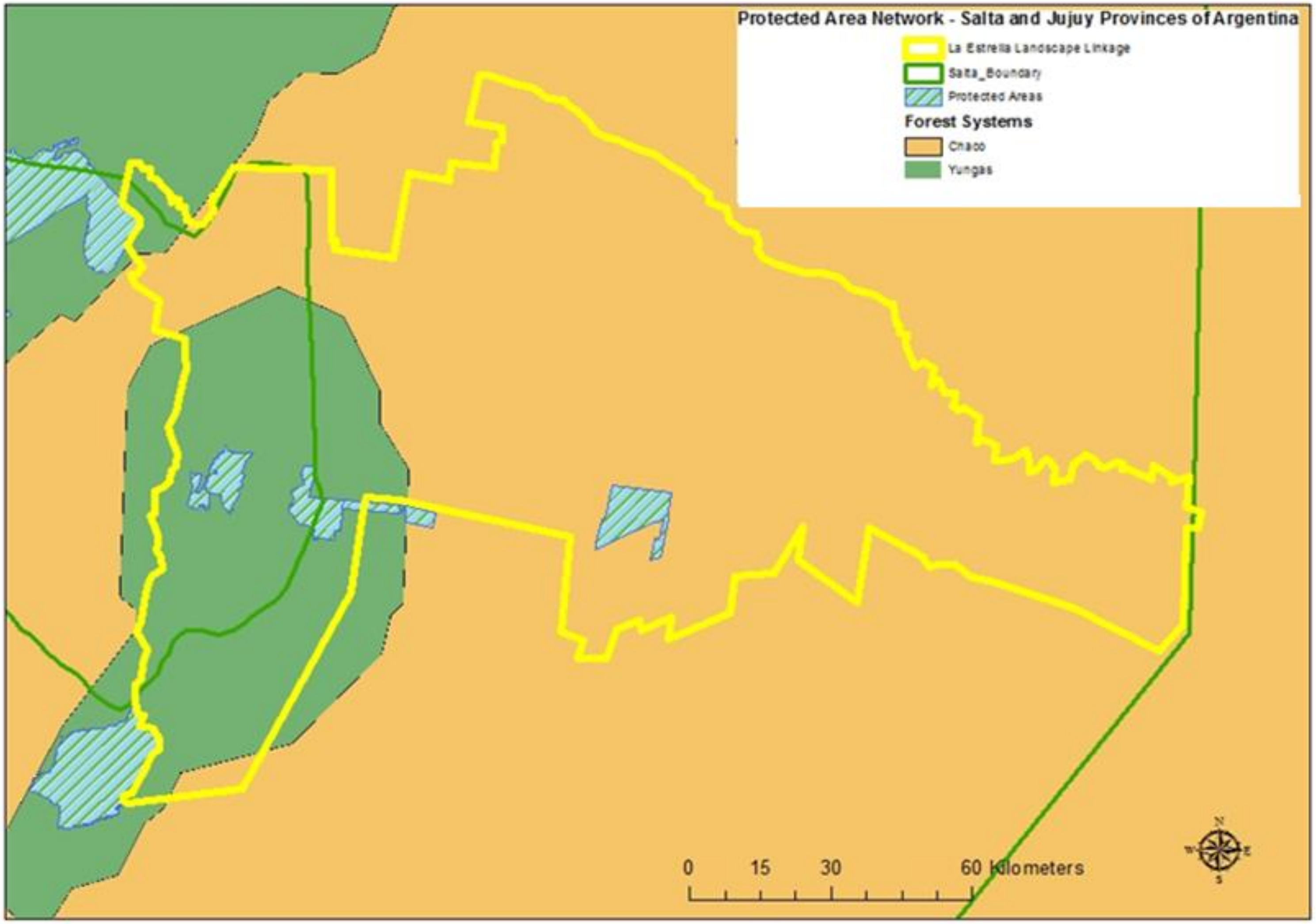

Figure 2: The study area, outlined in yellow, contains a network of provincial and national protected areas in both the Salta and Jujuy provinces, representing the Chaco and Yungas biomes. 2011.

(15,500 ha): Los Palmares and Las Lancitas. The western portion of the linkage is flanked by two national parks, El Rey National Park (Yungas-Chaco transition zone) along the southwestern edge, and Calilegua National Park (Yungas Forest) along the northwestern edge, consisting of 44,162 ha and 76,306 ha, respectively. A recently established national reserve, Pizarro (25,000 ha) and is located in the center of the study area, in the transition zone between the Yungas and Chaco forests (Fig. 2).

2.1.2 Road Network and Human Settlements

The entire study area contains an extensive road network that is particularly dense in its central and eastern sections. Past and current logging activities, and petroleum prospecting have contributed most to the network. A major paved interprovincial highway divides the central portion of the study area from north to south. Primary roads in the system are paved and secondary roads are consolidated gravel or dirt. These roads are relatively well maintained and wide (5m) compared to the tertiary road network. Tertiary roads in the study area are generally narrow (<5 m) and poorly maintained. Human use of the tertiary road network is lower compared to the primary and secondary road networks.

The study landscape is rural, located approximately 100 km from the provincial capitals of Salta and Jujuy. There are over 900 disbursed rural settlements (i.e., ranches or "puestos") most of which consist of 1-2 low income families, but some with up to seven households (Altrichter 2006). Ranch households in this region (i.e. "puesteros") generally subsist by grazing livestock such as cattle or goats (most often dispersed grazing), producing fuel wood, and hunting wild meat to complement their diets. Sixteen small towns (on average ≥20 households) are distributed throughout the study region (Altrichter and Boaglio 2004).

2.1.3 Hunting activity

When wild meat is consumed in the study region, mammals, including large mammals, constitute the main source of protein (Altrichter 2006, Di Bitetti et al. 2008). Even though wild meat is preferred in the average rural diet, domestic meat consumption may reflect the bulk of present day consumption (Altrichter 2006). According to hunter surveys conducted by Altrichter (2006) in rural areas proximate to the study area, 95% of rural peasants hunt game, actively or opportunistically while working in the forest or in agricultural fields. Hunting of large mammals is illegal in this region of Argentina (N. Politi, Fundación CEBIO, personal communication).

2.2 Data collection and processing

2.2.1 Data compilation and preparation

The majority of data compilation and preparation for this research was conducted using geographic information system (GIS) tools with ArcGIS v10 software (Environmental Systems Research Institute Inc., Redlands CA). We collected data on the location of human settlements, roads, and forest cover to spatially reference human disturbance that likely had the most important influence on large mammal mortality from hunting pressure (Trombulak and Frissell 2000, Seijas 2004, LaRue and Nielsen 2008, Rabinowitz and Zeller 2010, Sampaio et al. 2010, Schwartz 2010).

Based on Google Earth (2011), Landsat 5 (2010), and BingMaps satellite and aerial imagery covering the study area (Seijas 2004), we manually edited or newly digitized data layers (i.e., primary roads, tertiary roads, ranch settlements, and villages) using the ArcGIS ArcMap editing toolbar. Village locations were simply mapped as point features. We mapped additional village locations that were located within approximately 10 km of the study area, if they were within the range of influence of the study area and would impact our calculations of risk. To map locations of ranch settlements, we obtained a dataset detailing the locations of human structures from the National Parks Administration of Argentina (L. Lizarraga, GIS specialist, Salta, Argentina office). This dataset included geospatial information of the study area's various types of man-made structures including markets, schools, cemeteries, airports, and the ranch houses. In order to limit analysis to ranch sites only, we conducted a visual verification process supported by Google Earth technology (2011), to only include data points that appeared consistent with human habitation. Only those locations that included a house, water reservoir, and corrals, typically surrounded by bare soils due to vegetation degradation, were used as indicators of human habitation (Grau et al. 2008). As we encountered locations that appeared to be ranch houses that were not yet mapped, we digitized these as point features and added them into the dataset.

We again used satellite and aerial imagery to locate the study area's extensive road network using Google Earth (2011), Landsat 5 (2010), and BingMaps satellite and aerial imagery, and manually digitized each road segment as a line feature and classified roads as either primary or tertiary. As we digitized roads, we separated the two distinct road datasets from one another as their relative weights in our risk model were different. We bundled primary and secondary roads into the same category for purposes of this study. Perimeter roads that surround the edges of agricultural fields were included as a part of the tertiary road network because we assumed that all field perimeters permit easy access to forest edges in the form of a footpath or roadway. Altrichter (2006) reported that most rural peasants surveyed in locations near our study area actively sought game or engaged in opportunistic hunting while working in the forest or in agricultural fields.

We used Landsat 5 imagery from the growing season (March and April, 2010) to map percent forest cover. Estimates of forest cover were based on the Normalized Difference Vegetation Index (NDVI) which is an index of primary productivity (Pettorelli 2005).

NDVI = Near Infrared - Red / Near Infrared + Red

We used ENVI software to analyze and process geospatial imagery (ITT Visual Information Solutions) to calculate NDVI values using bands 3 and 4 of the Landsat images. We assumed that any NDVI value under 0.4 was not considered forested. The remaining values were stretched from 0 to 100% forest cover. Due to errors encountered with agricultural fields receiving artificially high NDVI values, all agricultural fields were manually masked in the study area and given a value of 0.0 for percent forest cover.

All data layers were standardized to the same projected coordinate system, WGS 1984 Web Mercator, and resampled to a 30 m grid to ensure the same resolution of analysis. We then converted these layers to raster format using ArcGIS, ArcToolbox- Conversion Tools, to enable subsequent geoprocessing steps (with the exception of the forest cover layer which was already in raster format; Seijas 2004).

2.2.2 Individual risk distribution models

To spatially model the association between large mammal mortality risk and proxies of human disturbance, we first estimated linear distances from four threat factors (Table 1) at which the influence of each threat factor declines to zero (Mockrin et al. 2011). That is, our model forces risk values towards a zero value with increasing distance from each threat source, following a normal distribution (Hill et al. 1997, Smith 2008). We extracted linear distance values that represent the spatial patterns of hunter travel distance when in search of large bodied mammals (Di Bitetti et al. 2008, Rabinowitz and Zeller 2010, Schwartz et al. 2010, Mockrin et al. 2011, Yackulic et al. 2011). After review of the range of distance values linked to hunter travel distance found in the literature, we selected distance values based upon the following general criteria (in order of priority): 1) proximity of reference site to study site; 2) relevance or similarity of reference site conditions to study site (e.g., habitat characteristics); and 3) distance values that were most recurrent (or overlapped approximately).

We assigned a 5 km impact radius zone around ranches, representing the average hunter travel distance from a homestead (Bonaudo et al. 2005, Altrichter 2006, Smith 2008). Village points were assigned a 16 km impact radius that represented the estimated distance hunters will enter surrounding forest adjacent to a village in search of game (Altrichter and Boaglio 2004, Sirén et al. 2004, Altrichter 2006, Peres and Nascimento 2006, Sarmiento 2007). We assigned a 2 km impact radius for both primary and tertiary roads (Franzen et al. 2006, Rabinowitz and Zeller 2010, Thoisy et al. 2010).

Table 1: 1997-2010. Neotropical study sites reporting hunting of large mammals by indigenous people or colonists. Distances traveled by hunters in search of game measured from points of human access and dwellings were recorded (i.e., roads, ranches, and villages).

Literature Reviewed			**Distances traveled by hunters in search of game measured from points of human access and dwellings (km)**			
Source	**Study Location**	**Species**[1]	**Distance from primary roads**	**Distance from tertiary roads**	**Distance from ranches**	**Distance from village**
Hill et al. (1997)	Eastern Paraguay	Large mammals	—[2]	6-10	—	—
Novaro et al. (2000)	Neotropics	Large mammals	—	—	—	10
Jerozolimski and Peres (2003)	Bolivia, Brazil, Colombia, Ecuador, Peru, Suriname, Venezuela	Large mammals	—	—	—	6-12
Naughton-Treves et al. (2003)	SE Peru	Large mammals	—	—	10	—
Peres and Lake (2003)	Brazil, Amazon Basin	Large mammals	5	5	9	9
Altrichter and Boaglio (2004)	Northern Argentina	*Tayassu pecari*	—	—	—	16
Sirén et al. (2004)	Eastern Ecuador	Large mammals	—	—	—	17
Bonaudo et al. (2005)	Brazilian Amazon	Large mammals	—	—	5	—
Altrichter (2006)	Chaco, Argentina	*Tayassu pecari*	—	—	5	5-100
Franzen et al. (2006)	Ecuadorian Amazon	Large mammals	—	3-4	—	—
Peres and Nascimento (2006)	SE Amazon Brazil	Large mammals	—	—	—	8-10 12-26
Sarmiento (2007)	Colombia	*Tapirus terrestris*	—	—	—	16
Smith (2008)	Western Panama	Large mammals		—	2-7	—
Parry et al. (2009)	NE Brazilian Amazon	Large mammals	—	—	—	10
Colchero et al. (2010)	SE Mexico	*Panthera onca*	1	1	—	—
Rabinowitz and Zeller (2010)	South America	*Panthera onca*	2	2	8	—
Thoisy et al. (2010)	French Guiana	Large mammals	2	2	—	—
Van Holt et al. (2010)	Bolivian Amazon	Large mammals	—	—	—	8

[1] Species listings that were too numerous for use in table, were labeled as "large mammals"
[2] No data

We wrote a program in Python (Appendix 1) to implement a normal distribution function on each of four threat factors (forest cover is dealt with separately). We developed a formula including three input variables to estimate risk for each threat factor (Table 2). Two of the model input variables required to support calculation of each threat factor's risk distribution were: 1) the impact radius defined as the linear distance travelled by a hunter in search of game from the origin of a threat source (for every point within the impact radius R ≥ 0); and 2) the relative weight assigned to each threat factor according to the estimated importance of each threat factor relative to the others (Seijas 2004).

Table 2: 2011. Risk distribution model parameters and formula applied to each threat factor within study area. Relative weights, impact radii, and variance used to describe model distribution for each threat factor.

Risk Distribution Model Formula	**Threat Factor**	**Weight**	**Impact Radius**[1] (km)	**Variance**
$R = W * e^{-\frac{1}{2}*(\frac{d}{V})^2}$	Villages	10	16	150
R= Risk value	Ranches	8	5	50
W= Weight	Primary Roads	8	2	10
d= Distance from threat factor	Tertiary Roads	6	2	10
V= Variance	Forest Cover[2]	4	—	—

[1] For d > impact radius, R= 0
[2] See section 2.2.2 for details

In general, low mortality risk locations for large mammals are locations far from villages, with low density of ranch settlements and roads, and high forest cover (Peres 2001, Altrichter and Boaglio 2004, Di Bitetti et al. 2008, Parry et al. 2009). We assigned relative weights to our individual risk distribution models (see below for detail on forest cover parameterization) to capture the relative influence that each threat factor is assumed to have on mortality risk for large mammals. Weights ranged from 0-10, with a 10 value equating to the highest risk possible in this model (Seijas 2004, Rabinowitz and Zeller 2010). We assigned the highest weight of 10 to villages, ranches and primary roads received a value of 8, and tertiary roads were weighted a value of 6 (Appendix 2; Altrichter and Boaglio 2004, Seijas 2004, Altrichter 2005, Altrichter 2006, Franzen et al. 2006, Rabinowitz and Zeller 2010, Thoisy et al. 2010).

The forest cover raster file was manipulated using raster calculator (ArcToolbox, Spatial Analyst, Map Algebra, Raster Calculator) to assign a relative weight to forest versus non-forest pixels. The high forest cover pixels received values close to 0 reflecting lower risk, while the low forest cover pixels received a value of 4, indicating relatively moderate risk from lack of hiding cover (Peres 2001, Naughton-Treves et al. 2003, Altrichter and Boaglio 2004, Parry et al. 2009).

Once we ran each individual risk distribution model calculation in Python, we applied our results in ArcMap (ESRI GIS) to spatially model each threat factor's distance function.

2.2.3 Creation of risk map- aggregation of all risk factors

We aggregated all risk distribution maps in ArcToolbox's Raster Calculator to produce the final risk map (Seijas 2004). Risk values summed over all maps ranged from 0-36, with 36 representing the highest predicted mortality risk for large mammals in this region.

3. RESULTS

Fifteen villages were mapped within the study area, and 5 villages were mapped outside the study area boundaries (Fig. 3). We mapped 916 ranch sites (Fig. 4). Primary roads were distributed most heavily in the west-central to western portion of the study area (Fig. 5). A dense network of tertiary roads dominated the entire study landscape, particularly its eastern section (Fig. 6).

Our final risk distribution maps for villages, ranches, primary roads, and tertiary roads (Figs. 7, 8, 9, and 10) show individual risk values that decline to zero at 16 km, 5 km, 2 km, and 2 km, respectively. While primary and tertiary roads were assigned the same impact radius (i.e, 2 km), the unique weights that were assigned to these risk factors generated distinct risk values (i.e., primary roads represented higher risk because they were weighted more heavily than tertiary roads). Our final risk distribution map for forest cover (Fig. 11) displayed gradients of risk that were not based on distance factors. Pixels that registered high forest cover (assumed to provide more hiding cover for mammals) registered as lower risk values (in cases of nearly complete forest cover, risk for this factor was given a "0" value), while areas of low forest cover (assumed to increase exposure and sightability of mammals) registered as relatively high risk values.

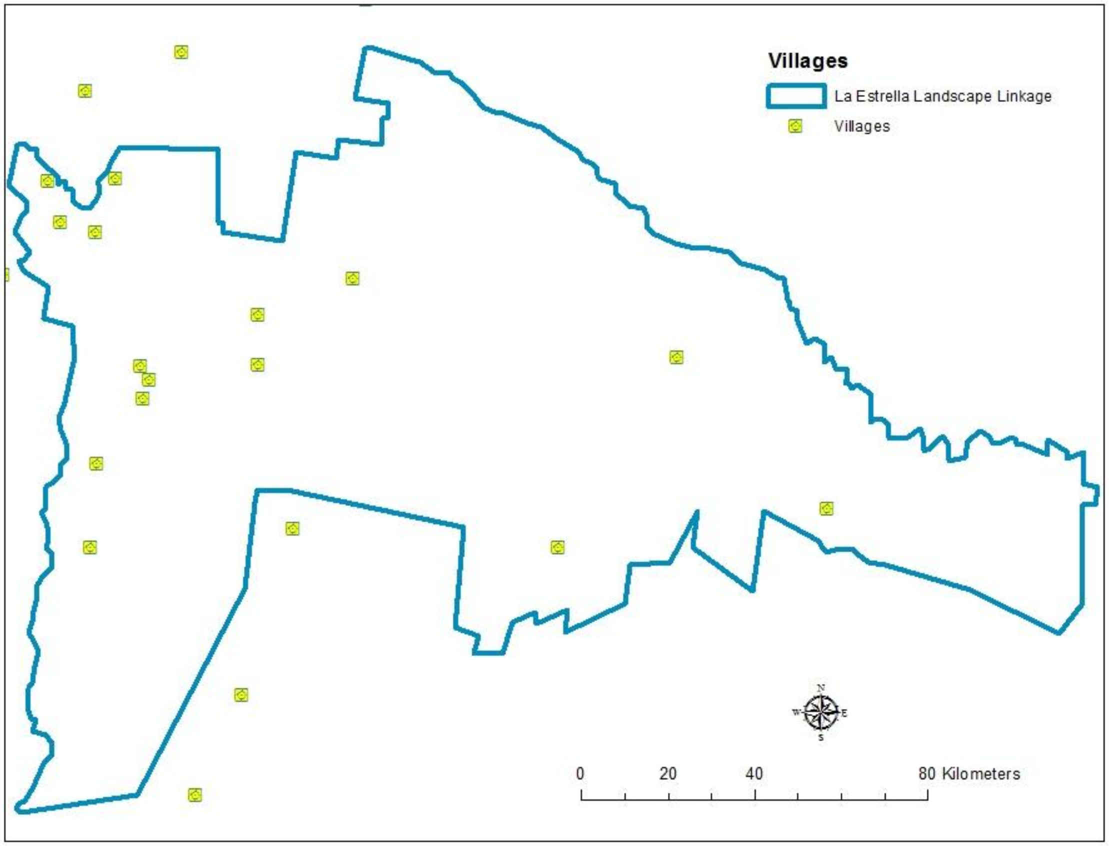

Figure 3: Study area, 2011. Villages within or proximate to study area.

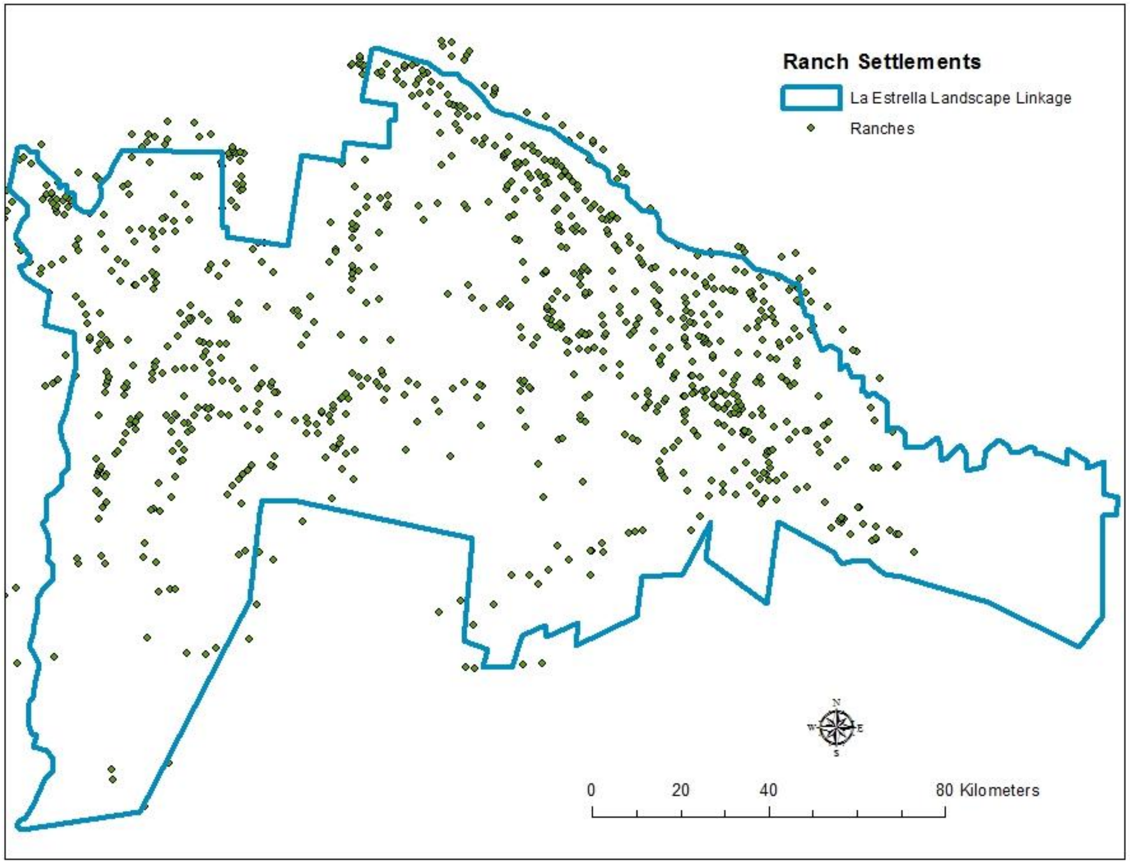

Figure 4: Study area, 2011. Ranch settlements within or proximate to study area.

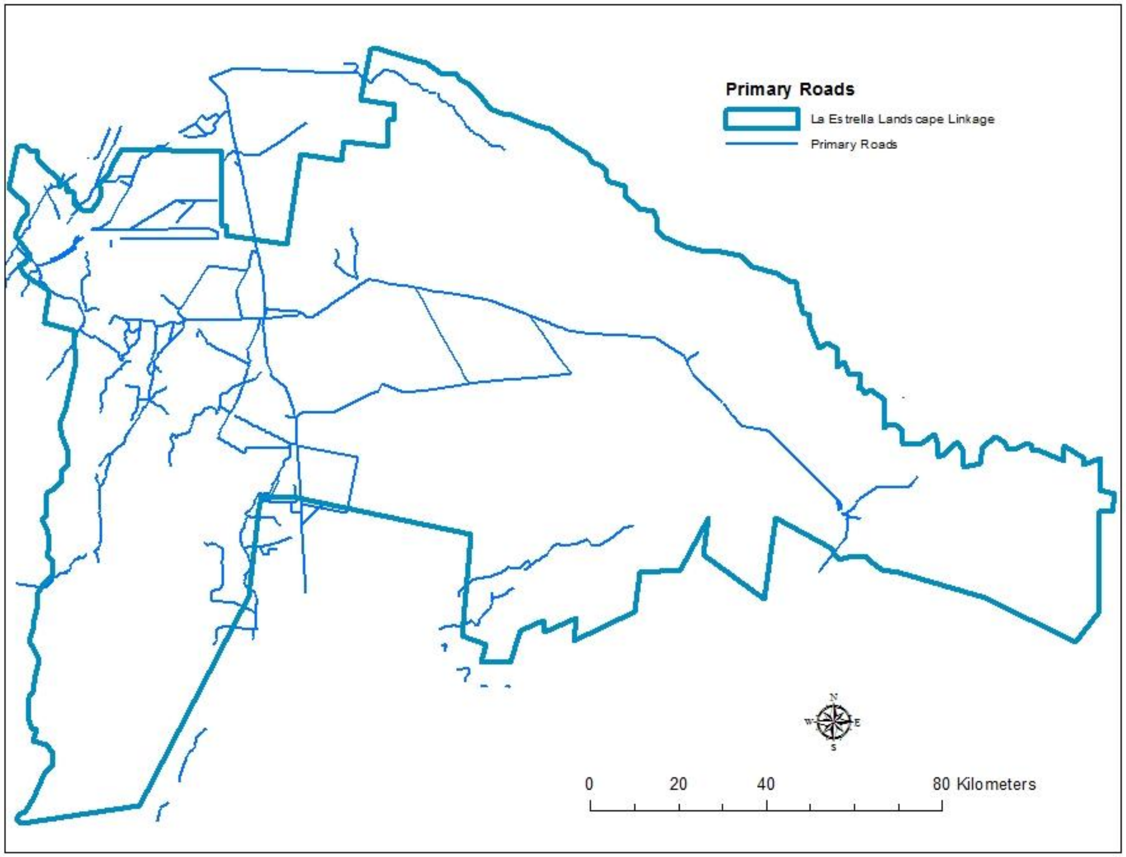

Figure 5: Study area, 2011. Primary and secondary road network.

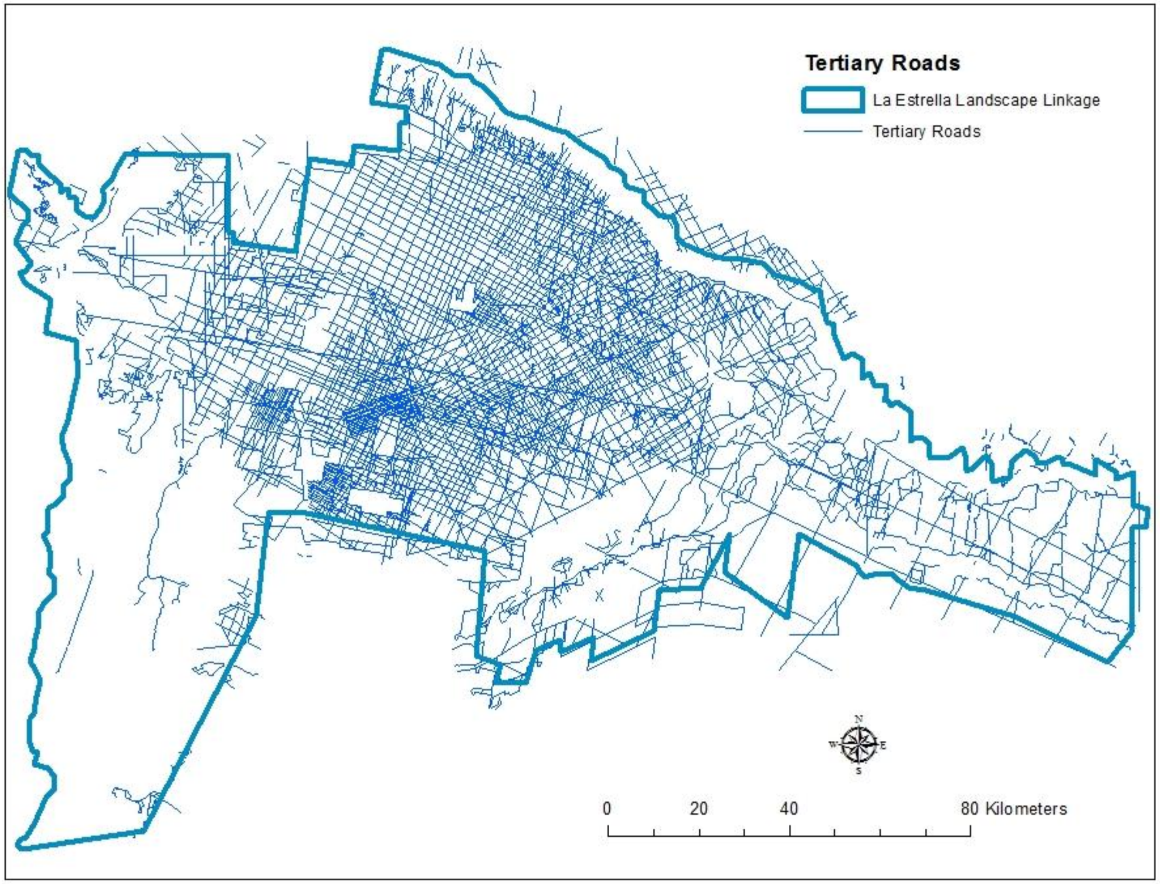

Figure 6: Study area, 2011. Tertiary road network.

Figure 7: Study area, 2011. Relative risk of human-caused mortality for large mammals as a function of distance from village locations.

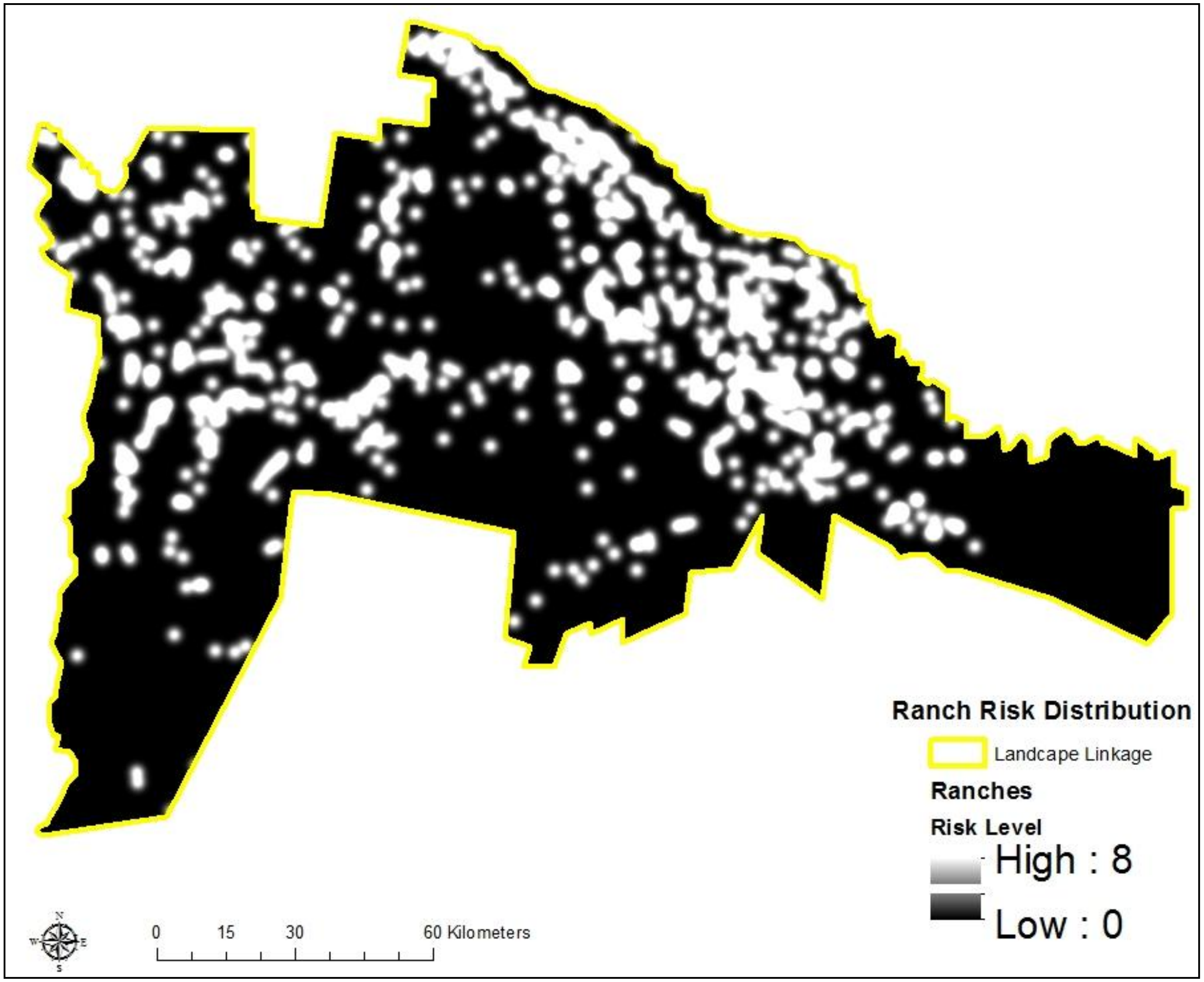

Figure 8: Study area, 2011. Relative risk of human-caused mortality for large mammals as a function of distance from ranch locations.

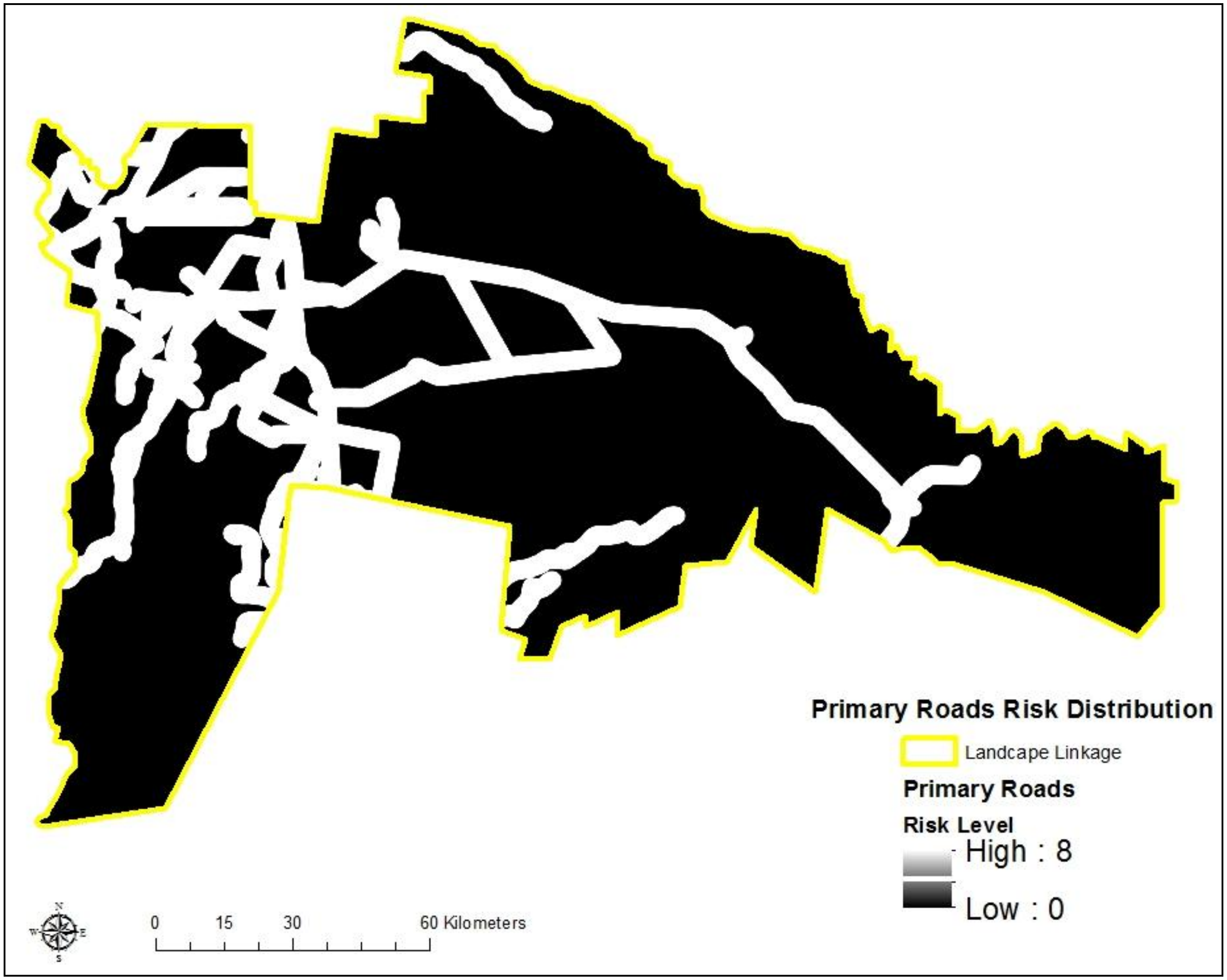

Figure 9: Study area, 2011. Relative risk of human-caused mortality for large mammals as a function of distance from primary roads.

Figure 10: Study area, 2011. Relative risk of human-caused mortality for large mammals as a function of distance from tertiary roads.

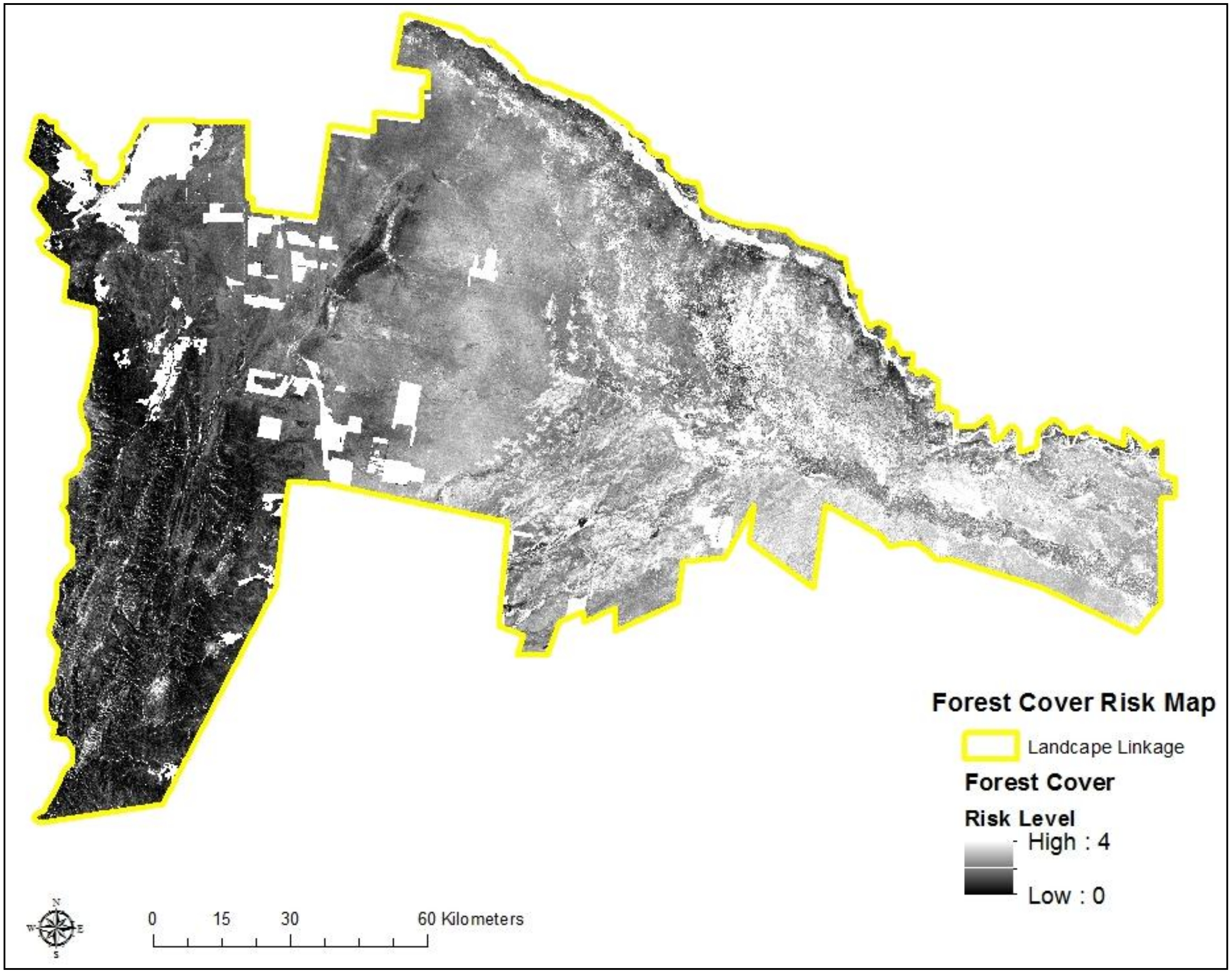

Figure 11: Study area, 2011. Relative risk of human-caused mortality for large mammals as a function of forest cover. Pixels shaded in darker grey represent more dense foliage while pixels in lighter grey indicate less forest cover.

A visual inspection of our final aggregated risk distribution map for large mammals (Fig. 12) showed that the majority of the study area's mortality risk values are within the upper half of possible risk values. Linear, high mortality risk values, dominated the model output, reflecting a confluence of threat factors found along primary roads. The central and eastern portions of the study area showed clear aggregations of ranch settlements, with an especially dense tertiary road network (Fig. 12). It appears the combined effect of tertiary roads and ranch settlements generated mid-range risk values throughout much of this portion of the study area. As shown in Fig. 12, forest cover in this area tended to have mid-range risk values contributing to the mid-range risk values distributed throughout this area. Results for the study area's western reach revealed overall risk distribution values that represent mid-range to high mortality risk for large mammals (particularly in the central and northern zones). The western side of the study area had a higher density of primary roads, a moderate density of tertiary roads, clustering of villages, and a moderately high number of ranch settlement locations. Forest cover was higher on this side of the study area, however there were more agricultural clearings distributed throughout much of this territory.

Our final aggregated risk distribution map overlaid with provincially planned land clearing zones (Fig. 13; Ordenamiento Territorial de Bosques Nativos N° 7543, Salta; Ordenamiento Territorial Adaptativo para Áreas Boscosas N°5676, Jujuy) revealed substantial areas slated for agricultural clearing that fell within low to mid-range risk values. Intervening protected areas within the study landscape (Fig. 13) had risk value outputs that were surprisingly elevated considering their land use designation.

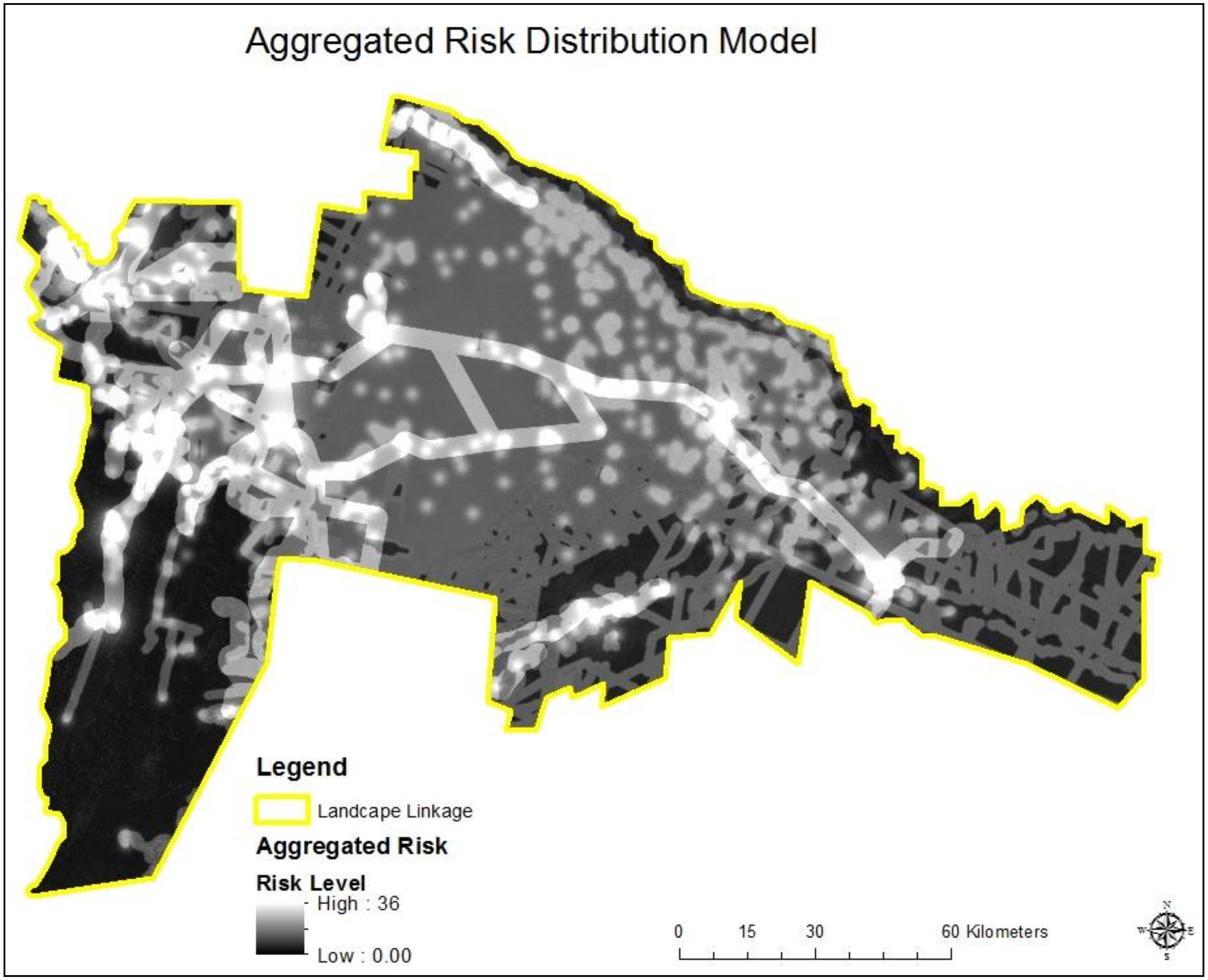

Figure 12: Study area, 2011. Final aggregated map showing relative risk of human-caused mortality to large mammals as a function of the distribution of anthropogenic threat factors. Each pixel of this map was classified according to a gradient of risk ranging from 0 to 36, reflecting the sum of risk values from all individual risk distribution models.

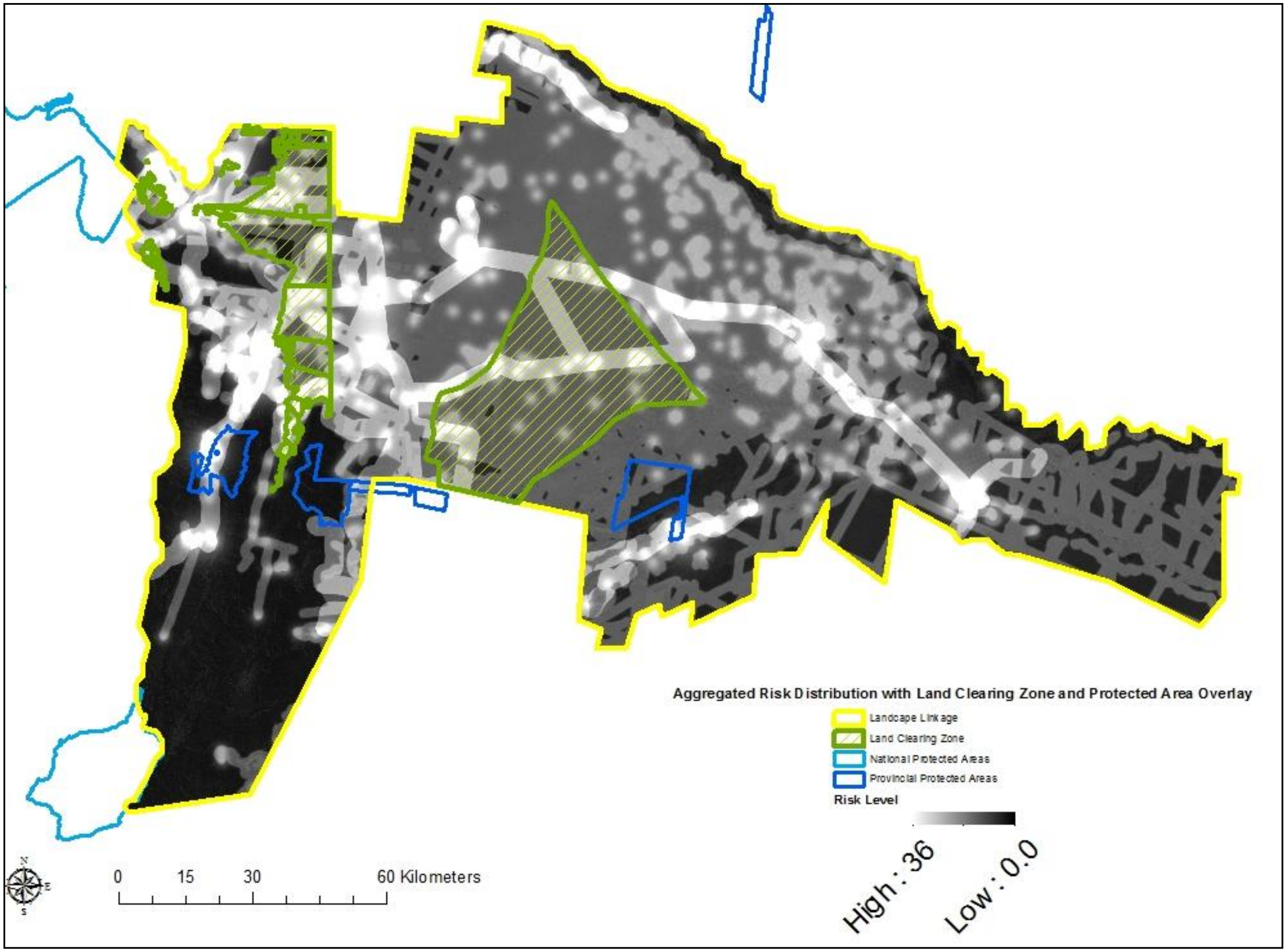

Figure 13: Study area, 2011. Final aggregated map showing relative risk of human-caused mortality to large mammals as a function of the distribution of anthropogenic threat factors, overlaid by provincially-authorized land clearing zones and national and provincial protected areas.

4. DISCUSSION

Illicit hunting of large mammals remains a common practice in northwestern Argentina. Our model represents the first spatial approximation of mortality risk due to hunting pressure on large mammals. Our results have revealed relatively high levels of mortality risk for large mammals that potentially inhabit or move through the study region. Increased fragmentation may limit the ability of hunted populations to be replenished by less impacted populations located in the region (Novaro et al. 2000, Altrichter 2005). These effects can be exacerbated by improved human access and hunting methods. In the case of the study area, many hunters have improved means of hunting efficiency (e.g., shotguns, cars, motorcycles) which may allow them to harvest a larger numbers of individuals per hunting event, and reach locations further from settlements. Because overall mortality risk distributed throughout the landscape was approximately moderate-high (particularly in the central to eastern sections, and the northwestern section), increased modes of hunter efficiency may have caused some instances of underestimation of large mammal mortality risk in the study area.

The mortality risk distribution model for human-caused hunting of large mammals presented here provided analysis for large mammals that was general in nature, from studies across Latin America. Thus, this model does not capture the variability of potential influences that each anthropogenic threat factor has on individual species (De Angelo et al. 2011). Despite potential risk, not all large mammals are deterred from locations of anthropogenic disturbance, such as roads. Species such as brocket deer (*Mazama americana*), ocelot (*Leopardus pardalis*), grey brocket deer (*Mazama gouazoubira*), pampas fox (*Pseudalopex gymnocercus*), and Chacoan cavy (*Pediolagus salinicola*) use roads for reasons including ease of movement (Di Bitetti et al. 2008), which could in fact put them at risk in locations that are more densely developed. Some species are more tolerant to hunting pressure than other species that have lower reproductive rates or other harvest sensitivities (Bodmer et al. 1997, Di Bitetti et al. 2008, De Angelo et al. 2011). In fragmented forest landscapes such as in the study area, animals that occupy larger home ranges and generally occur at low densities, including jaguar, may be particularly affected by potential hunting pressure if they are forced to cross several human hunting sites to meet their life history requirements; thus exposing them to higher risk of mortality (Chiarello 1999, De Angelo et al. 2011). At the same time, animals that are not as highly mobile (such as the giant anteater and collared peccary) may be unable to adequately respond to localized hunting pressure and avoid mortality risk posed by human hunting pressure (Mockrin et al. 2011). In areas where there is less protection from hunting pressure, large mammals may be forced to shift their activity patterns to crepuscular or nocturnal hours, to minimize the potential for encounters with humans (Paviolo et al. 2009). These shifts in activity patterns have unknown consequences for species survival rates.

In the study region, hunters are likely to differ in their approaches to hunting which is partly dependent on their mode of transportation and experience in forested landscapes (Table 1; Mockrin et al. 2011). Hunters may select more remote, heavily forested locations with limited access, in search of bigger game (Parry et al. 2009, Mockrin et al. 2011). Due to the extensive distribution of Chaco forest (also known as the "Impenetrable" for its thorn forests and dense shrubbery) in the study area, we may have overestimated hunter accessibility along the primary and tertiary road network. Local knowledge of the area suggests that accessibility from roads into most forested locations is less than the 2 km value we conservatively assigned to our model, due to the dense forest type; hence hunters may only be able to access locations immediately off the road in many areas (pers. com. Luis Rivera, Fundación CEBio). While our model did not incorporate variation in hunter activity patterns, it captures the cumulative effects of risk of mortality posed by hunting accessibility in this system (Yackulic et al. 2011).

Risk of large mammal mortality attributed to hunting in our study landscape has the potential to be expansive and intense enough to trigger unsustainable hunting conditions for large mammals. Evidence from this study and other studies that have been conducted in the region (Grau and Brown 2000, Altrichter and Boaglio 2004, Altrichter 2006, Altrichter et al. 2006, Ojeda et al. 2008, Chalukian et al. 2009) suggests that large mammals in the study region are in danger of extirpation. This situation necessitates a response from scientists and managers to advance research initiatives to better understand species harvest rates and wildlife population dynamics in the area. Yet conducting detailed species inventories in the field is severely constrained by limited resources and time, which may not allow managers to keep pace with planned land clearing activities. Substantial areas in the study landscape are slated for agricultural clearing (Fig. 13) coinciding with areas of low to mid-range mortality risk for large mammals. If development (including associated roads and human settlements) proceeds in these areas of overlap, large mammal mortality risk levels will increase, potentially increasing hunting pressure. Intervening protected areas within the study landscape (Fig. 9) had risk value outputs in some locations that were surprisingly elevated considering their land use designation. New development in, around, and between the existing protected area network, if poorly planned, may jeopardize the relatively low risk values that currently describe the majority of land within these protective boundaries.

The pressures facing the forested biomes of northwestern Argentina are mounting. Given that wildlife harvest is operating at unsustainable levels throughout Latin America (Bodmer et al. 1997, Peres 2001), and evidence from this study suggests a similar situation could develop in the study region, provincial and local authorities along with conservation biologists should refocus attention and resources to this issue and apply precautionary principals directly toward essential anti-poaching campaigns and public outreach programs. If the persistence of wildlife is valued by the region's stakeholders, then reevaluation of planned land clearing activities is recommended before direct connectivity between the Yungas and Chaco forests is eliminated, and increased development pressure introduces new and elevated mortality risks to large mammals attempting to persist in this region. Managers may use the results of this study to help delineate priority locations for initial conservation action, and as future studies regarding wildlife population dynamics and harvest become available, more informed management plans may be generated.

Large forest vertebrates hold tremendous importance in the Neotropics for their role in ecological and social functioning, and merit immediate conservation action due to their high susceptibility to overhunting (Redford 1992, Di Bitetti 2008, Paviolo et al. 2009, Sampaio et al. 2010). As most of the land in the study region is privately held, the extension of conservation efforts to private lands is critical to preventing extirpation of the region's wildlife. Effective conservation of wildlife biodiversity along the deforestation frontier of northwestern Argentina will require increased research, policy coordination, and the strategic support of private land interests and local communities.

REFERENCES

Altrichter, M. and Boaglio, G.I., 2004. Distribution and relative abundance of peccaries in the Argentine Chaco: associations with human factors. Biological Conservation. 116, 217-225.

Altrichter, M, 2005. The sustainability of subsistence hunting of peccaries in the Argentine Chaco. Biological Conservation. 126, 351-362.

Altrichter, M. 2006. Wildlife in the life of local people of the semi-arid Argentine Chaco. Biodiversity and Conservation. 15, 2719-2736.

Altrichter, M., Boaglio, G. and Perovic, P., 2006. The decline of jaguars *Panthera onca* in the Argentine Chaco. Oryx. 40, 302-309.

Bodmer, R., Eisenberg, J., Redford, K.,1997. Hunting and the likelihood of extinction of Amazonian mammals. Conservation Biology. 11, 460-466.

Bodmer, R.E., Robinson, J.G., 2004. Evaluating the sustainability of hunting in the Neotropics, in: Silvius, K.M., Bodmer, R.E., Fragoso, J.M.V. (Eds.), People in Nature. Columbia University Press, New York, pp. 299-323.

Boix, L. R. and Zinck, J.A., 2008. Land-use planning in the Chaco plain (Burruyacu, Argentina). Part 1: evaluating land-use options to support crop diversification in an agricultural frontier area using physical land evaluation. Environmental Management. 42,1043–1063.

Bonaudo, T., Pendu, Y.L., Faure, J.F., Quanz, D. 2005. The effects of deforestation on wildlife along the transamazon highway. European Journal of Wildlife Resources. 51,199-206.

Brown A. D., Grau, H. R., Malizia L. R. & Grau, A., 2001. Los bosques nublados de la Argentina, in: Bosques Nublados del Neotrópico. Kapelle, M., Brown, A.D. (Eds.). Editorial INBio, Costa Rica, pp. 623-659.

Chalukian, S., de Bustos, M. S., Lizárraga, L., Varela, D., Paviolo, A., Juliá, J. P. and Quse, V., 2009. Plan de acción para la conservación del tapir (*Tapirus terrestris*) en Argentina. Wildlife Conservation Society, Tapir Specialist Group-UICN, Dirección de Fauna – Secretaría de Ambiente y Desarrollo Sustentable de la Nación. http://www.ambiente.gov.ar/archivos/web/Tapir/file/Plan_de_Acci%C3%B3n_Tapir_Final.pdf

Chiarello, A., 1999. Effects of fragmentation of the Atlantic forest on mammal communities in south-eastern Brazil. Biological Conservation. 89, 71-82.

Colchero, F., Conde, D.A., Manterola, C., Chávez, C., Rivera, A., Ceballos, G., 2010. Jaguars on the move: modeling movement to mitigate fragmentation from road expansion in the Mayan Forest. Animal Conservation. 1-9.

Cullen Jr., L., Bodmer, R., Padua, C., 2000. Effects of hunting in habitat fragments of the Atlantic forests, Brazil. Biological Conservation. 95, 49-56.

Cushman, S.A., McKelvey, K.S., Hayden, J., Schwartz, M.K., 2006. Gene flow in complex landscapes: testing multiple hypotheses with causal modeling. The American Naturalist. 168, 486-499.

De Angelo, C., Paviolo, A., Di Bitetti, M. 2011. Differential impact of landscape transformation on pumas (*Puma concolor*) and jaguars (*Panthera onca*) in the Upper Paraná Atlantic Forest. Diversity and Distributions. 17, 422-436.

Di Bitetti, M., A. Paviolo, C. Ferrari, C. De Angelo, Y. Di Blanco. 2008. Differential responses to hunting in two sympatric species of brocket deer (*Mazama americana and M. nana*). Biotropica. 40, 636-645.

Franzen, M., 2006. Evaluating the sustainability of hunting: a comparison of harvest profiles across three Huaorani communities. Environmental Conservation. 33, 36-45.

Gardner, R.H., Gustafson, E.J., 2004. Simulating dispersal of reintroduced species within heterogeneous landscapes. Ecological Modelling. 171, 339-358.

Gasparri, N. I., Grau, H. R., 2009. Deforestation and fragmentation of Chaco dry forest in NW Argentina (1972-2007). Forest Ecology and Management. 258, 913-921.

Grau, A., Brown, A. 2000. Development Threats to Biodiversity and Opportunities for Conservation in the Mountain Ranges of the Upper Bermejo River Basin, NW Argentina and SW Bolivia. Ambio. 29, 445-450.

Grau, H. R., Gasparri, N. I., Aide,T.M., 2005. Agriculture expansion and deforestation in seasonally dry forests of north-west Argentina. Environmental Conservation. 32, 140–148.

Hill, K., Padwe, J.,Bejyvagi, C., Bepurangi, A., Jakugi, F., Tykuarangi, R., Tykuarangi, T.,1997. Impact of hunting on large vertebrates in the Mbaracayu Reserve, Paraguay. Conservation Biology. 11, 1339-1353.

Jerozolimski, A., Peres, C.A., 2003. Bringing home the biggest bacon: a cross-site analysis of the structure of hunter-kill profiles in Neotropical forests. Biological Conservation. 111, 415-425.

LaRue, M. A., Nielsen, C.K., 2008. Modelling potential dispersal corridors for cougars in Midwestern North America using least-cost path methods. Ecological Modelling. 212, 372-381.

Mills, L.S., Allendorf, F.W., 1996. One-migrant-per-generation rule in conservation and management. Conservation Biology. 10, 1509-1518.

Mockrin, M.H., Rockwell, R.F., Redford, K.H., Keuler, N.S., 2011. Effects of landscape features on the distribution and sustainability of ungulate hunting in northern Congo. Conservation Biology. 25, 514-525.

Naughton-Treves, L., Mena, J.L., Treves, A., Alvarez, N., Radeloff, V.C., 2003. Wildlife survival beyond park boundaries: the impact of slash-and-burn agriculture and hunting on mammals in Tambopata, Peru. Conservation Biology. 17, 1106-1117.

Noss, A. and Cuéllar, R.L., 2008. La sostenibilidad de la cacería de Tapirus terrestris y de Tayassu pecari en la tierra comunitaria de origen isoso: el modelo de cosecha unificado. Mastozoología Neotropical. 15, 241-252.

Novaro, A.J., Redford, K.H., Bodmer, R.E., 2000. Effect of hunting in source-sink systems in the Neotropics. Conservation Biology. 14, 713-721.

Ojeda, R. A., Stadler, J., Brandl, R.. 2003. Diversity of mammals in the tropical-temperate Neotropics: hotspots on a regional scale. Biodiversity and Conservation. 12, 1431–1444.

Ojeda, R. A., Barquez, R.M., Stadler, J., Brandl, R., 2008. Decline of mammal species diversity along the Yungas Forest of Argentina. Biotropica. 40, 515-521.

Parry, L., Barlow, J., Peres, C.A., 2009. Allocation of hunting effort by Amazonian smallholders: implications for conserving wildlife in mixed-use landscapes. Biological Conservation. 142, 1777-1786.

Paviolo, A., Di Blanco, Y., De Angelo, C., Di Bitetti, M., 2009. Protection affects the abundance and activity patterns of pumas in the Atlantic Forest. Journal of Mammalogy. 90, 926-934.

Peres, C.A., 2001. Synergistic effects of subsistence hunting and habitat fragmentation on Amazonian forest vertebrates. Conservation Biology. 15, 1490-1505.

Peres, C.A., Lake, I.R., 2003. Extent of nontimber resource extraction in tropical forests: accessibility to game vertebrates by hunters in the Amazon Basin. Conservation Biology. 17, 521-535.

Pettorelli, N., Vik, J.O., Mysterud, A., Gaillard, J., Tucker, C.J., Stenseth, N.C., 2005. Using the satellite-derived NDVI to assess ecological responses to environmental change. Trends in Ecology and Evolution. 20, 503-510.

Rabinowitz, A., Zeller, K.A., 2010. A range-wide model of landscape connectivity and conservation for the jaguar, *Panthera onca*. Biological Conservation. 143, 939-945.

Rosati, V.R., Bucher, E.H., 1995. Relative abundance and diet composition of Chacoan cavies in relation to range condition. Journal of Range Management. 48, 482-486.

Redford, K.H., Taber, A., Simonetti, J.A., 1990. There is more to biodiversity than the tropical rain forests. Conservation Biology. 4, 328-330.

Redford, K.H., 1992. The empty forest. BioScience. 42, 412-422.

Reyna-Hurtado, R., Naranjo, E., Chapman, C.A., Tanner, G.W., 2010. Hunting and the conservation of a social ungulate: the white-lipped peccary Tayassu pecari in Calakmul, Mexico. Oryx. 44, 89-96.

Roldán, A., Simonetti, J., 2001. Plant-mammal interactions in tropical Bolivian forests with different hunting pressures. Conservation Biology. 15, 617-623.

Sampaio, R., Lima, A.P., Magnusson, W.E., Peres, C.A., 2010. Long-term persistence of midsized to large-bodied mammals in Amazonian landscapes under varying contexts of forest cover. Biodiversity and Conservation. 19, 2421-2439.

Sarmiento, A.M., 2007. Patrones en la distribución de los lugares de captura del tapir (*Tapirus terrestris*) con base en el conocimiento tradicional de las comunidades indígenas Andoque y Nonuya, y el asentamiento de Puerto Santander-Araracuara, Amazonia Colombiana. Thesis. National University of Colombia, Bogotá.

Schwartz, C., Haroldson, M., White, G., 2010. Hazards affecting grizzly bear survival in the Great Yellowstone Ecosystem. Journal of Wildlife Management. 74, 654-667.

Seijas, A.E., 2004. Abundance, Spatial Distribution, and Human Pressure on Orinoco Crocodiles (*Crocodylus intermedius*) in the Cojedes River System, Venezuela, in: Silvius, K.M., Bodmer, R.E., Fragoso, J.M.V. (Eds.), People in Nature. Columbia University Press, New York, pp. 227-239.

Sirén, A., Hamback, P., Machoa, J., 2004. Including spatial heterogeneity and animal dispersal when evaluating hunting: a model analysis and an empirical assessment in an Amazonian Community. Conservation Biology. 18, 1315-1329.

Smith. N. J. H., 1976. Utilization of game along Brazil's transamazon highway. Acta Amazonica. 6, 455-466.

Smith, D.A., 2008. The spatial patterns of indigenous wildlife use in western Panama: implications for conservation management. Biological Conservation. 141, 925-937.

Sowls, L.K., 1997. Javelinas and other peccaries: their biology, management, and use. Texas A&M Press, College Station, USA.

Tabeni, M.S., Bender, J.B., Ojeda, R.A., 2004. Puntos calientes para la conservación de mamíferos en la Provincia de Tucumán, Argentina. Mastozoología Neotropical. 11, 55-67.

Taber, A., Chalukian, S., Altrichter, M., Minkowski, K., Lizárraga, L., Sanderson, E., Rumiz, D., Ventincinque, E., Moraes Jr., E., de Angelo, C., Antúnez, M., Ayala, G., Beck, H., Bodmer, R., Boher, S., Cartes, J., de Bustos, S., Eaton, D., Emmons, L., Estrada, N., de Oliveira, L., Fragoso, J., Garcia, R., Gomez, C., Gómez, H., Keuroghlian, A., Ledesma, K., Lizcano, D., Lozano, C., Montenegro, O., Neris, N., Noss, A., Vieira, J., Paviolo, A., Perovic, P., Portillo, H., Radachowsky, J., Reyna-Hurtado, R., Ortiz, J., Salas, L., Duenas, L., Perea, J., Schiaffino, K., de Thoisy, B., Tobler, M., Utreras, V., Varela, D., Wallace, R., Ríos, G., 2008. El destino de los arquitectos de los bosques neotropicales: evaluacion de la distribucion y el estado de conservacion de los pecaries labiados y los tapires de tierras bajas. Grupo Especialista de la CSE/UICN en Cerdos, Pecaríes y Hipopótamos; Grupo Especialista de la CSE/UICN en Tapires; Wildlife Conservation Society; and Wildlife Trust. New York, NY, USA.

Talamo, A., Caziani, S.M., 2003. Variation in woody vegetation among sites with different disturbance histories in the Argentine Chaco. Forest Ecology and Management. 184, 79-92.

Thoisy, B., Richard-Hansen, C., Goguillon, B., Joubert, P., Obstancias, J., Winterton, P., Brosse, S., 2010. Rapid evaluation of threats to biodiversity: human footprint score and large vertebrate species responses in French Guiana. Biodiversity Conservation. 19, 1567-1584.

Trombulak, S.C., Frissell, C.A., 2000. Review of ecological effects of roads on terrestrial and aquatic communities. Conservation Biology. 14, 18-30.

Van Holt, T., Townsend, W.R., Cronkleton, P., 2010. Assessing local knowledge of game abundance and persistence of hunting livelihoods in the Bolivian Amazon using consensus analysis. Human Ecology. 38, 791-801.

Wilkie, D.S., Bennett, E.L., Peres, C.A., Cunningham, A.A., 2011. The empty forest revisited. Annals of the New York Academy of Sciences. 1223, 120-128.

Yackulic, C.B., Strindberg, S., Maisels, F., Blake, S., 2011. The spatial structure of hunter access determines the local abundance of forest elephants (*Loxodonta Africana cyclotis*). Ecological Applications. 21, 1296-1307.

APPENDIX 1

Python script developed by Adam Moreno, Numerical Terradynamic Simulation Group, University of Montana, 4/13/2011

```
*****************************************************
"""
Purpose:   Creates impact map with normal distribution dampening
           radiating from point.  There are confounding effects
           resulting from the overlapping of impact from different
           points.  If the resulting impact is over a maximum weight
           then it equals the maximum weight.
Input:     Raster images with points that will create impact
output:    Raster image of resulting spatial impacts
"""
#*****************************************************


import numpy
import math
from osgeo import gdal                    #import geospatial data analysis module
from osgeo.gdalconst import *
from osgeo import gdalconst
import numpy as np


gdal.AllRegister()
sampleMapPath = '/net/orthanc/USRangelands/miroc32_a1b_7_09/shrubsLAISum'
outputmapPath = './PrimaryRoads6KMfasterdieout10a'
inputRaster = './PrimaryRoads'

mapFile1 = gdal.Open( sampleMapPath, GA_ReadOnly )      #open ENVI file
if mapFile1 is None:                          #test to see if we we could open file
   print 'Could not open ' + sampleMapPath
   sys.exit(1)
mapFile = gdal.Open( inputRaster, GA_ReadOnly )      #open file
if mapFile is None:                           #test to see if we we could open file
   print 'Could not open ' + mapPath
   sys.exit(1)
rows  = mapFile.RasterYSize
cols  = mapFile.RasterXSize

rasterGridBand = mapFile.GetRasterBand(1)
rasterGrid1 = rasterGridBand.ReadAsArray(0,0,cols, rows)
rasterGrid = np.zeros([rows, cols], float)

distance = 6000 m
weight = 8.0
pixelDistance = int(distance/30)
rasterFlagValue = 8.0
variance = 10.0 #higher = slower die down lower means steeper
print rows, cols
print "distance in meters = ", distance, "km = ", pixelDistance, " pixels"
for r in range (0,rows):
   for c in range (0, cols):
      if rasterGrid1[r][c] == rasterFlagValue:
         #print "Inside", rasterGrid[r][c], r, c
         #raw_input()
         rLeft = r-pixelDistance
         rRight = r + pixelDistance
         cUp = c - pixelDistance
         cDown = c + pixelDistance
         #print rLeft, rRight, cUp, cDown
         #raw_input()
         for rL in range(rLeft, rRight):
            for cU in range (cUp, cDown):
               #print rL, cU, rLeft, rRight, cUp, cDown
               if rL < rows and cU < cols and rL > 0 and cU > 0:
```

```
                        if rasterGrid1[rL][cU] != rasterFlagValue:
                            ra = math.sqrt((math.fabs(r-rL)**2)+(math.fabs(c-cU)**2))
                            #print rL, cU, ra, rRight, cDown
                            rasterGrid[rL][cU] += float(weight * math.exp((-1/2)*((ra/variance)**2)))
                            #print rasterGrid[rL][cU]
                            #raw_input()
                            if rasterGrid[rL][cU] >= weight:
                                rasterGrid[rL][cU] = weight
                        else:
                            rasterGrid[rL][cU] = weight
    #print r, rows

mapFile1 = gdal.Open( sampleMapPath, GA_ReadOnly )       #open ENVI file
if mapFile1 is None:                                     #test to see if we we could open file
    print 'Could not open ' + sampleMapPath
    sys.exit(1)
driver = mapFile1.GetDriver()
RasterImageFile = driver.Create(outputmapPath, cols, rows, 1, gdalconst.GDT_Float32)

newBand = RasterImageFile.GetRasterBand(1)
newBand.WriteArray(rasterGrid,0,0)
newBand.FlushCache()
```

APPENDIX 2

Visual representation of individual risk distribution model.

<table>
<tr><th>Relative Risk Distribution Model</th><th>Threat Factor</th><th>Weight</th><th>Impact Radius (km)</th><th>Variance</th></tr>
<tr><td rowspan="5">$$R = W * e^{-\frac{1}{2}*\left(\frac{d}{V}\right)^{2}}$$
R= Risk value

W= Weight

d= Distance from threat factor

V= Variance</td><td>Villages</td><td>10</td><td>16</td><td>150</td></tr>
<tr><td>Ranches</td><td>8</td><td>5</td><td>50</td></tr>
<tr><td>Primary Roads</td><td>8</td><td>2</td><td>10</td></tr>
<tr><td>Tertiary Roads</td><td>6</td><td>2</td><td>10</td></tr>
<tr><td>Forest Cover</td><td>4</td><td>–</td><td>–</td></tr>
</table>